Short-Term Vegetation Response Following Mechanical Control of Saltcedar (*Tamarix* spp.) on the Virgin River, Nevada, USA

Steven M. Ostoja, Matthew L. Brooks, Tom Dudley, and Steven R. Lee*

Tamarisk (a.k.a. saltcedar, *Tamarix* spp.) is an invasive plant species that occurs throughout western riparian and wetland ecosystems. It is implicated in alterations of ecosystem structure and function and is the subject of many local control projects, including removal using heavy equipment. We evaluated short-term vegetation responses to mechanical *Tamarix* spp. removal at sites ranging from 2 to 5 yr post-treatment along the Virgin River in Nevada, USA. Treatments resulted in lower density and cover (but not eradication) of *Tamarix* spp., increased cover of the native shrub *Pluchea sericia* (arrow weed), decreased density and cover of all woody species combined, increased density of both native annual forbs and the nonnative annual *Salsola tragus* (prickly Russian-thistle), and lower density of nonnative annual grasses. The treated plots had lower mean woody species richness, but greater herbaceous species richness and diversity. Among herbaceous species, native taxa increased in richness whereas nonnative species increased in both species richness and diversity. Thus, efforts to remove *Tamarix* spp. at the Virgin River reduced vegetative cover contributing to fuel loads and probability of fire, and resulted in positive effects for native plant diversity, with mixed effects on other nonnative species. However, absolute abundances of native species and species diversity were very low, suggesting that targets of restoring vegetation to pre-invasion conditions were not met. Longer evaluation periods are needed to adequately evaluate how short-term post-treatment patterns translate to long-term patterns of plant community dynamics.

Nomenclature: *Tamarix* (tamarisk).

Key words: Virgin River, saltcedar, riparian, invasive species control, bull dozer.

Riparian ecosystems contain critical natural resources, buffer anthropogenic contaminants, and stabilize stream channels (Naiman et al. 1993; Sabo et al. 2005). Most riparian areas have been highly modified through alterations to natural flow regimes, water diversions and ground water pumping (Richardson et al. 2007) often resulting in a suite of negative ecological effects (Patten 1998, Stromberg 2001). In riparian zones of the arid and semi-arid western United States, one of the primary indicators of altered conditions is the presence and dominance of nonnative invasive plants, chiefly *Tamarix* spp. (tamarisk, saltcedar) (Hultine and Dudley 2013).

Intentionally introduced to North America from southern Europe, Asia, and North Africa, *Tamarix* spp. has been used for windbreaks and to resist streambank erosion. Escaping cultivation by the 1930s, it spread rapidly throughout western riverine systems coincident with the expansion of water development programs (Billington et al. 2005; Robinson 1965). *Tamarix* spp. has colonized and in many cases now dominates numerous river ecosystems, springs, and seeps, including relatively unregulated systems, replacing native vegetation in large areas of the arid western United States (Friedman et al. 2005).

Tamarix spp. invasion is associated with a suite of ecosystem impacts including increased water consumption, altered river channel form, increased soil salinity and degraded wildlife habitat quality (Bateman and Ostoja 2012; Shafroth et al. 2005). Controlling *Tamarix* spp. has, therefore, been a major element of local, state, and federal invasive species management programs (Shafroth et al. 2010). Mechanical cutting or uprooting and removal of *Tamarix* spp., generally with large-scale herbicide application, are commonly used control methods involving a variety of techniques ranging from crews with power or hand tools to heavy equipment extraction (i.e., bulldozers)

DOI: 10.1614/IPSM-D-13-00064.1

* First, second and fourth authors, Ecologist, Research Botanist, and Botanist, respectively, U.S. Geological Survey, Western Ecological Research Center, Yosemite Field Station, Oakhurst California, USA, 93644; third author: Research Biologist, Marine Science Institute, University of California, Santa Barbara, California, USA 93106-6150. Corresponding author's E-mail: sostoja@usgs.gov

Management Implications

Short-term reductions in density and cover of *Tamarix* spp. can be achieved with mechanical control techniques, as applied on the Virgin River, Nevada, USA. These treatments can also reduce total woody plant cover and may provide some reduction in flammable nonnative annual grass density, but have no net effect on total herbaceous density. Thus, treatments can reduce fuelbeds, and the potential for wildfire where fuel reduction is the principal goal of vegetation management. Nonnative annual forbs, *Salsola tragus* (Russian-thistle) in particular, can increase indicating that control treatments for *Tamarix* spp. can facilitate secondary invasion by other nonnative species. Treatments can also increase cover of rhizomatous native woody plants such as *Pluchea sericea* (arrowweed), and native forbs as a group, as well as increase diversity for various plant guilds. Treatments provided no significant benefit to arboreal native plants such as *Prosopis* spp. (mesquites) nor mesic taxa including *Salix* spp. (willows) and *Populus fremontii* (Fremont cottonwood) that may be of greater value to wildlife. The results of this short-term response study are the foundation for understanding longer-term (i.e. 5+ yr) *Tamarix* spp. control efficacy or plant community trajectories, and provide a basis to assess how long the initial treatment effects may persist. In addition, it is unknown how the results of mechanical treatments evaluated in this study compare with those of other approaches (e.g. herbicide, fire, biological control, or combinations thereof). Both short and long term effects, and the relative pros and cons of various control techniques, should be considered when developing any *Tamarix* spp. management plan.

(see O' Meara et al. 2010). However, limited information exists regarding the relative effectiveness of these and other control methods (c.f. Bay and Sher 2008; Harms and Hiebert 2006).

There is also concern that highly manipulative and aggressive control techniques may have unintended negative consequences in the form of increased bare soil resulting in soil erosion, the short-term loss of native vegetation and/or vertical habitat structure, and subsequent facilitation of secondary invasions by other nonnative species (D'Antonio and Meyerson 2002; Shafroth et al. 2008). Management depends on monitoring of vegetation responses following removal to assess the degree to which control efforts may facilitate short and long-term regeneration of native species. With the recent introduction of biological control for *Tamarix* spp. suppression (Bean et al. 2012), it becomes particularly relevant to evaluate the effectiveness of current weed management practices and how future management practices could be enhanced.

In this study we evaluated short-term vegetation responses to *Tamarix* spp. control using mechanical techniques along the Virgin River in southern Nevada, USA (Dudley and Brooks 2011). We hypothesized that: (1) *Tamarix* spp. density and cover are lower in treated than untreated sites (to quantify the effect of the control treatments on the target); (2) native woody plant density, cover, and species diversity are lower in the treated than untreated sites (because natives were removed along with *Tamarix* spp.); and (3) herbaceous plant density and species diversity (both native and nonnative) are greater in the treated than untreated sites. We also evaluated species composition within treated areas to assess patterns that may help to focus management actions designed to promote dominance of native over nonnative species.

Materials and Methods

Study Site. The study area was located along the Virgin River approximately 11 to 14 km (6.8 to 8.7 mi) downriver (southwest) from Mesquite, NV (36°44′18.12″N 114°12′17.47″W). The Virgin River is only moderately regulated and retains a seminatural flood regime, typically with year-round surface flows. *Tamarix* species in this area included the saltcedars, *T. ramosissima* Ledeb., *T. chinensis* Lour. and their interspecific hybrids, and *T. parviflora* DC. (smallflower tamarisk). Information provided by the Bureau of Land Management (BLM), Las Vegas Field Office indicated that treated areas were mechanically cleared of all *Tamarix* spp. and other above-ground plant material, and vegetation biomass was removed off-site, between 2004 and 2007. These records were not sufficient to determine the specific year each area was treated, so we were unable to consider time as a factor in analyses.

Vegetation removal was accomplished using heavy equipment according to standard BLM operating procedures. The approximate percentage of removal methods across all the treated areas was 60% mastication, 25% dozer, 14% chainsaw/hand tools, and 1% excavator/extraction (T. Rash, personal communication). Some of the treated areas were planned for follow-up herbicide treatments, but records were insufficient to determine when and where they were done. Field observations indicated that the treated sites were also scraped and experienced significant soil disturbance (S. Ostoja, personal observation). Untreated areas were in locations with no record of *Tamarix* spp. removal and were interspersed among treated areas along a 35 km reach of the Virgin River through the Virgin Valley (from near Bunkerville to approx. 10 km downstream of Riverside, NV). Based on our observations along treated and nearby areas, we surmised that *Tamarix* spp. cover was fairly uniform and $> 70\%$ across the entire study area.

Study Design. Using Hawth's analysis tools in ArcGIS 9.1 (ESRI, Redlands, CA) we created a grid of 250 by 250 m (820 ft) (62,500 m^2 (672,400 sq ft) or 6.25 ha) (15.5 ac)cells and overlaid it on the treated and nearby untreated areas. This resulted in 75 total gridded cells or macroplots, 45 in treated areas and 30 in untreated areas. The 6.25 ha macroplot size was selected based on the appropriate sampling scale for bird community measurements associated with a companion study which will be published elsewhere (preliminary results in Dudley and Brooks 2011), but it also served as a suitable replicate unit to evaluate vegetation responses. The macroplots serve as the unit of replication in the current study.

Vegetation was subsampled within each replicate macroplot using 5 by 30 m (150 m^2) vegetation plots established in spring 2009. Vegetation sampling was, therefore, done 2 to 5 yr after *Tamarix* spp. control treatments were implemented (2004 through 2007). Three vegetation plots were randomly located within each block in the treated areas (3 times 45 macroplots = 135 total vegetation plots) and two were randomly located within each block in the untreated areas (2 times 30 macroplots = 60 total vegetation plots). More vegetation plots were placed within treated compared to untreated areas because initial observations indicated that plant community heterogeneity was much greater in those areas due to control activities than in untreated areas. Species nomenclature, native status, life history (annual, perennial), and growth habitat (forb, grass/monocot, shrub, tree) were derived from the U.S. Department of Agriculture plants database (www.plants.usda.gov).

Density, cover and species identity of all woody plants (shrubs and trees) were measured within each 150 m^2 vegetation plot. Each individual plant having > 50% of its rooted base within the plot was counted. Cover of woody perennial plants was measured by the point–intercept method using one of the 30 m sides of the vegetation plot as a transect. Starting at the end of each transect and repeated every 30 cm (11.8 in), a 0.65 cm-diam sampling rod oriented perpendicular to the ground was extended vertically and each hit of the rod tip with vegetation was recorded by species. Since the transect length was 30 m, there were 100 points from 30 to 3,000 cm.

Density and species identity of all herbaceous plants (annual forbs, annual grasses, perennial forbs, and perennial grasses/monocots) were recorded within three 1 m^2 subsampling frames at the beginning, middle, and end of each of the two 30 m sides of the vegetation plot, resulting 6 subsampling frames per vegetation plot. Density of herbaceous plants was determined by counting plants with any part of their rooted base within the subsampling frame.

Data Analyses. Density and cover of woody plants were averaged across vegetation plots within each 6.25 ha replicate macroplot. Density of herbaceous plants was averaged across the 1 m^2 subsampling frames within each of the 150 m^2 vegetation plots then averaged across the vegetation plots within each replicate macroplot. In this way, replicate macroplots were treated as the random factor in all analyses. We square root transformed count (density) data and for proportional data (cover) we used the arcsine to better meet normality assumptions for statistical comparisons, but in all cases present back-transformed data in figures. Analyses were performed in JMP 8.0.1 (SAS 2009).

To determine site diversity (alpha diversity) for each treatment condition, we calculated species richness, the Brillouin evenness index, and two species diversity indices, Simpson's D and Shannon–Wiener. Woody plant diversity was at the scale of the vegetation plot (150 m^2) and herbaceous plant diversity was at the scale of the subsampling frame (1 m^2). Species richness was calculated as the number of species per unit area, also referred to as species density. The Brillouin evenness index is scaled between 0 and 1 and assumes that all species are represented within the sample; larger values suggest a more even community. Simpson's D index represents the probability that two randomly selected individuals are of the same species. It takes into account both the number of species and their relative abundances, and more heavily weights the more abundant species. The Shannon–Weiner index is influenced by both the number of species and the evenness of species abundances; both a greater number of unique species and a more even species distribution increase this index. For both indices, larger values indicate greater diversity. Diversity and evenness indices were calculated using Species Diversity & Richness 4.1.2 Software (Seaby and Henderson 2006). We then used randomization tests to compare species diversity and evenness between treatment types (Solow 1993). A randomization test calculates the diversity index for, in this case, the treated and untreated sites and the difference between these indices (delta) is stored. Then 10,000 random assignments and calculation of delta are undertaken. The observed value of delta is compared against the observed distribution of delta values generated at random to determine if the observed value for the difference between the indices of the two samples (i.e., treated and untreated sites) could have been generated by random chance (Solow 1993). These analyses were considered significant at $\alpha = 0.05$ (Seaby and Henderson 2006).

Multivariate patterns of species presence among treatment plots were visualized using nonmetric multidimensional scaling (nMDS). Species abundances were square root transformed, standardized using the Wisconsin double standardization technique, and ordinated using Bray Curtis dissimilarity distances. A complete linkage cluster analysis was performed on the dissimilarity matrix and significant clusters ($\alpha = 0.05$) were identified using a similarity profile permutation test (SIMPROF). Analyses were done with vegan (Oksanen et al. 2012) and clustsig (Whitaker and Christman 2010) packages in the R statistical software environment, version 2.14.2 (R Core Team 2012).

Results and Discussion

Tamarix spp. Density and Cover Response. Treated areas had a 91% reduction in *Tamarix* spp. density (df = 1.73; $F = 7.28$; $P < .0086$) (Figure 1A) and a 96% reduction in

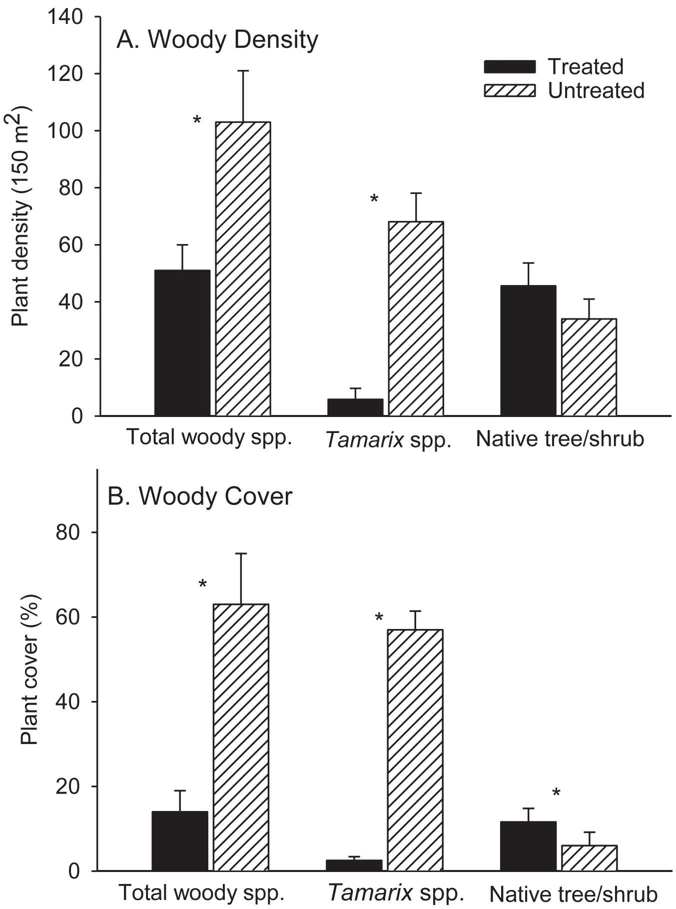

Figure 1. Panel A. Woody density for all woody species combined, *Tamarix* spp. and native trees and shrubs for the treated (black bars) and untreated (hashed bars) sites. Panel B. Woody cover (%) for all woody species combined, *Tamarix* spp. and native trees and shrubs for the treated (black bars) and untreated (hashed bars) sites. Within group comparisons denoted by '*' indicate statistically significant differences, refer to the text for specific statistical values for each test.

Tamarix spp. cover (df = 1.73; F = 657.3; P < .0001) (Figure 1B). These results corroborate another retrospective study spanning 1 to 11 post-treatment yr that included Mojave Desert riparian systems showing that both cut-stump and burn methods could reduce *Tamarix* spp. cover 89 to 99% (Harms and Hiebert 2006). Other programs have shown mechanical removal treatments to be similarly effective (Douglass et al. 2013). However, the fact that *Tamarix* spp. in treated areas was still 9% of the density and 4% of the cover found in untreated areas indicates that the treatments did not result in eradication of the target species.

Lack of complete eradication of target invasive species is typical of one-time invasive species control efforts, and follow-up treatments over time are typically recommended (e.g., Mack and Foster 2009). In the case of *Tamarix* spp., initial control treatments often focus on reducing biomass by mechanical removal methods such as those used in the current study, but also including root-plowing, prescribed fire and other control approaches (Douglass et al. 2013; Hart et al. 2005; Horton and Campbell 1974; Shafroth et al. 2005; Taylor and McDaniel 1998). Follow-up treatments typically involve herbicide application to resprouting stems, and retreatments for multiple years until no resprouting stems or emerging seedlings remain (McDaniel and Taylor 1999). Follow-up treatment may also incorporate the seeding or planting of other species to compete with and suppress the target nonnatives (Shafroth et al. 2008).

Native Woody Plant Density and Cover Response. We hypothesized that both density and cover of native woody species would be lower in treated than untreated areas due to the indiscriminate effects of the treatment methods on all woody plants, particularly because there had been relatively little time post-treatment for recruitment of native species. However, the results were mixed. Density of native trees and shrubs did not significantly differ between the treated and untreated sites (df = 1.73; F = 0.599, P = 0.441) (Fig 1A). In contrast, cover of native shrubs and trees was 45% higher in treated than untreated sites (df = 1.73; F = 5.73; P = .0193) (Figure 1B). These patterns are largely driven by a single native species, *Pluchea sericea* (Nutt.) Coville, that may not provide high quality wildlife habitat because stands of this species exhibit low structural diversity (Bateman and Ostoja 2012). *Pluchea sericea* is a relatively low-growing (< 2m), shrubby species that spreads vegetatively in open areas from shallow rhizomes so its abundance was characteristic of an early-successional riparian assemblage.

Harms and Hiebert (2006) report that mean native cover was approximately 176% greater in areas where *Tamarix* spp. was removed compared to where it was not removed, a response that was over 3× stronger than that found in the current study. This difference is likely due to their longer post-treatment span (1 to 11 yr) compared to the 2- to 5-yr span evaluated in the current study. It may also be an indication that that native propagules were more abundant prior to treatment, or surviving rootstock may have occurred in greater abundance following treatments, in the Harms and Hiebert (2006) study compared to the sites in the current study, Native arboreal riparian species are relatively sparse in the Virgin River floodplain where the current study was conducted, with *Populus fremontii* S. Watson (Fremont cottonwood) and *Salix gooddingii* C.R. Ball (Goodding's willow) absent from major sections of the river (Dudley and Bean 2012). In such circumstances, resilience of native species may be low and active restoration may be required to supplement recovery of native trees to sustainable population sizes.

Total Woody Density and Cover Response. Density of all woody plants was reduced by 50% (df = 1.73; F = 9.86; P = .0024) (Figure 1A) and cover was reduced by 22% (df = 1.73; F = 303.9; P < .0001) (Fig 1B), in treated compared to untreated areas. This difference in cover compares to the Harms and Hiebert (2006) retrospective study which reported that reduction in total vegetative cover (woody and herbaceous combined) after *Tamarix* spp. control ranged from 75 to 83%. Although it was not a specific focus of this study, we also measured percent cover of plant litter on the soil surface and found that it too was reduced by 33% in treated (37% cover) compared to untreated (55% cover) areas (df = 1,73; F = 109.01; P < 0.001). Although dense vegetation, often characteristic of *Tamarix* spp. stands, can promote sediment deposition, it can also lead to narrowing of the stream channel and subsequent increased rates of erosion (Friedman et al. 1996). However, removal of *Tamarix* spp. poses other threats of erosion since the loss of vegetation and litter cover is often of concern because of the erosion potential of desert riparian soils (Jaeger and Wohl 2011; Parsons et al. 1996). For example, a flood on the Rio Puerco in New Mexico USA removed 680,000 m^3 of sediment following *Tamarix* spp. control (Vincent et al. 2009). This is relevant for the Virgin River watershed with its high sediment transport rates (greater than 4 million metric tons annually) and episodic erosion/deposition and major channel evulsion events in this segment of the river (Hilmes and Vaill 1997).

Loss of vegetative cover and physiognomic structure following aggressive mechanical control of *Tamarix* spp. may also degrade wildlife habitat (Dudley and Brooks 2011), particularly for avian species, which can depend more on vegetation structure than species composition (Fleishman et al. 2003; Jones and Bock 2005; Sogge et al. 2005). Both cover as reported in the current study, and vegetation structure as reported in a separate report (Dudley and Brooks 2011), were negatively affected by the same *Tamarix* spp. control treatments in our study area. On the other hand, reduction in woody cover substantially reduced fuel loads at treatment sites, and lower *Tamarix* spp. cover and biomass is associated with reduced wildfire intensity and spread (Drus et al. 2012). Fuels reduction was the primary objective of the *Tamarix* spp. control treatments in this ecosystem, but lowering fire risk can also benefit wildlife, including the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) which has experienced nest failures in this and other regional watersheds as a result of wildfires fueled by *Tamarix* spp. (Dudley and Bean 2012; Marshall and Stoleson 2000). In the absence of *Tamarix* spp. control, periodic fires facilitate the incremental replacement of the mixed native/nonnative woodland that provides good quality wildlife habitat (van Riper et al. 2008) by a self-promoting near-monoculture of *Tamarix* spp. (Drus 2012) as seen in the majority of the lower Virgin River despite its relatively natural and minimally regulated hydrologic regime (Mortenson and Weisberg 2010).

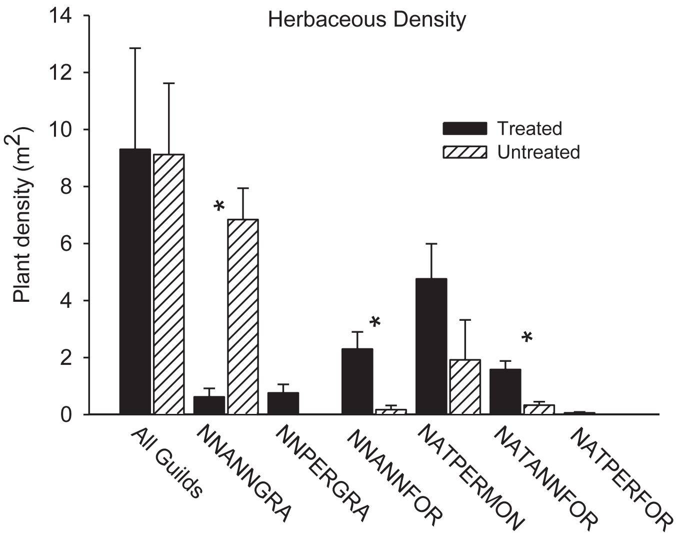

Figure 2. Herbaceous density for all species/guilds combined, nonnative annual grasses (NNANNGRA), nonnative perennial grasses (NNPERGRA), nonnative annual forbs (NNANNFOR), native perennial monocots (NATPERMON) and native perennial forbs (NATPERFOR) for the treated (black bars) and untreated (hashed bars) sites. Within group comparisons denoted by '*' indicate statistically significant differences, refer to the text for specific statistical values for each test.

Herbaceous Density Response. We hypothesized that herbaceous plant density would be greater in the treated sites due to competitive release after woody plants were removed, but we found no overall difference between the treated and untreated plots (df = 1.73; F = 0.256; P = 0.614) (Figure 2). We did find significant responses within individual plant guilds, although they did not all trend in the same direction. Specifically, density of nonnative annual grasses (NNANNGRA) was 91% lower in treated than untreated areas (df = 1.73; F = 8.05; P = 0.006) (Figure 2). In contrast, density of nonnative annual forbs (NNANNFOR) was 93% higher (df = 1.73; F = .256; P = 0.614), and density of native annual forbs (NATANNFOR) was 81% higher (df = 1.73; F =5.01; P = 0.028), in treated compared to untreated areas (Figure 2). Thus, mechanical removal of *Tamarix* spp. promoted the dominance of some secondary nonnative invaders, but suppressed others. It is possible that the herbaceous species are competing amongst themselves, and that their initial densities and habitat suitability for annual grasses versus forbs, may be reasons for the observed plant community patterns. *Salsola* spp., a major nonnative component of this forb complex, is often a superior competitor with grasses under dry, low-nutrient conditions (Allen 1982), particularly owing to its early germination and rapid taproot growth in disturbed sandy soils, and high water use efficiency (Beckie and Francis 2009; Young 1991).

Native perennial grasses (NATPERGRA) (df = 1.73; F = 2.22; P =0.140) and native annual forbs (NATANNFOR) (df = 1,73; F = 1.02; P = 0.317) did not differ between the treated or untreated plots. Nonnative perennial grass density (NNPERGRA) was 0.76 m^2 (± 0.24 SE) in the treated plots; however, no species within this group was detected within any untreated vegetation plot. Mean density of native perennial forbs (NATPERFOR) was 0.06 m^2 (± 0.03 SE) in the treated sites; however, no individuals were detected in the untreated sites (see Figure 1). These density values are very low and may be reflective of a generally depauperate condition of native plant communities in the Virgin River study area.

In a study designed to evaluate the effects of removing *Tamarix* spp. and *Elaeagnus angustifolia* L. (Russian-olive) in Canyon de Chelly National Monument, AZ, cut-stump and whole-plant removal resulted in decreased cover of nonnative herbaceous species and increased cover of native herbaceous species (Reynolds and Cooper 2011). Our contrary results seem to indicate that the treatment methods used were not effective in yielding substantial enhancement of native plant species, particularly herbaceous plants that contribute to understory diversity. The intensive soil disturbance associated with *Tamarix* spp. removal treatments could be a factor inhibiting native forb recruitment, damaging or deeply burying the existing seed banks. Native recruitment is often poor during low-flow drought years, as the higher terraces occupied by *Tamarix* spp. are infrequently scoured by flows that promote seed deposition and germination. In addition, it is likely that 'legacy effects' from many years of *Tamarix* spp. domination have contributed to making sites unsuitable for native establishment, either from chemical constituents that inhibit germination (e.g. salinization of soils) or by diminishing the microbial assemblage that facilitates growth and establishment of native plants in these sandy, low-nutrient soils (Meinhardt and Gehring 2013) .

Species Richness and Diversity. Of the 24 perennial woody species detected, 21 occurred in treated plots and 13 in untreated plots. Of the 38 herbaceous species detected (including annual forbs, annual grasses, perennial forbs, and perennial grasses/monocots categories), 33 occurred in treated plots and 16 in untreated plots. These differences translated into significantly higher mean (i.e., average/plot) species diversity in treated than untreated areas for the following guilds: mean perennial woody species (Shannon Wiener), mean herbaceous species (species richness, Simpsons D, Shannon Wiener, Brillouin Evenness), mean native herbaceous species (species richness), and mean nonnative herbaceous species (species richness, Simpsons D, Shannon Wiener, Brillouin Evenness) (Table 1). In only one case was a measure of diversity higher in untreated areas, and that was for mean woody species richness (Table 1).

Table 1. Simpsons and Shannon Wiener diversity indices and the Brillouin evenness index by treatment type for (A) woody species, (B) total herbaceous species, (C) native herbaceous species and (D) nonnative herbaceous species. Species richness values were determined by averaging the number of species detected by subplot for the treated and untreated plots. Significant results (α = 0.05) for randomization tests are indicated with bold font of the condition with the higher mean value (Solow 1993). One-way ANOVA of woody species richness was significantly greater in the untreated plots (df = 1.73; F = 4.34; P = 0.0405). One-way ANOVA of species richness for herbaceous species was greater in the treated plots (df = 1.73; F = 41.51; P < 0.001). One-way ANOVA of species richness for native herbaceous species was greater in the treated plots (df = 1.73; F = 26.89; P < 0.001) and was greater for nonnative herbaceous species as well (df = 1.73; F = 16.85; P < 0.001).

Guild index	Treated	Untreated
A) Woody species		
Species richness (mean)	1.93 (0.14)	**2.24 (0.18)**
Simpsons D	3.21	2.32
Shannon Wiener	**1.95**	1.28
Brillouin Evenness	0.62	0.48
B) Total herbaceous species		
Species richness (mean)	**2.38 (0.15)**	0.84 (0.18)
Simpsons D	**13.78**	5.11
Shannon Wiener	**2.92**	1.99
Brillouin Evenness	**0.83**	0.71
C) Native herbaceous species		
Species richness (mean)	**0.98 (0.08)**	0.27 (0.10)
Simpsons D	7.48	4.94
Shannon Wiener	2.34	1.76
Brillouin Evenness	0.78	0.79
D) Nonnative herbaceous species		
Species richness (mean)	**1.26 (0.11)**	0.53 (0.13)
Simpsons D	**4.73**	2.73
Shannon Wiener	**1.91**	1.29
Brillouin Evenness	**0.76**	0.56

Higher diversity following invasive plant control treatments suggests that opening of the canopy, as well as breaking up the matted litter layer, reduced the apparent competitive exclusion of other plant species from pretreatment *Tamarix* spp. stands and facilitated their establishment either from pre-existing seedbanks and vegetative structures or from dispersal of propagules into cleared sites. Both shading and inhibition of germination by litter are factors known to promote competitive dominance by *Tamarix* spp., although pre-emptive inhibition mechanisms are the

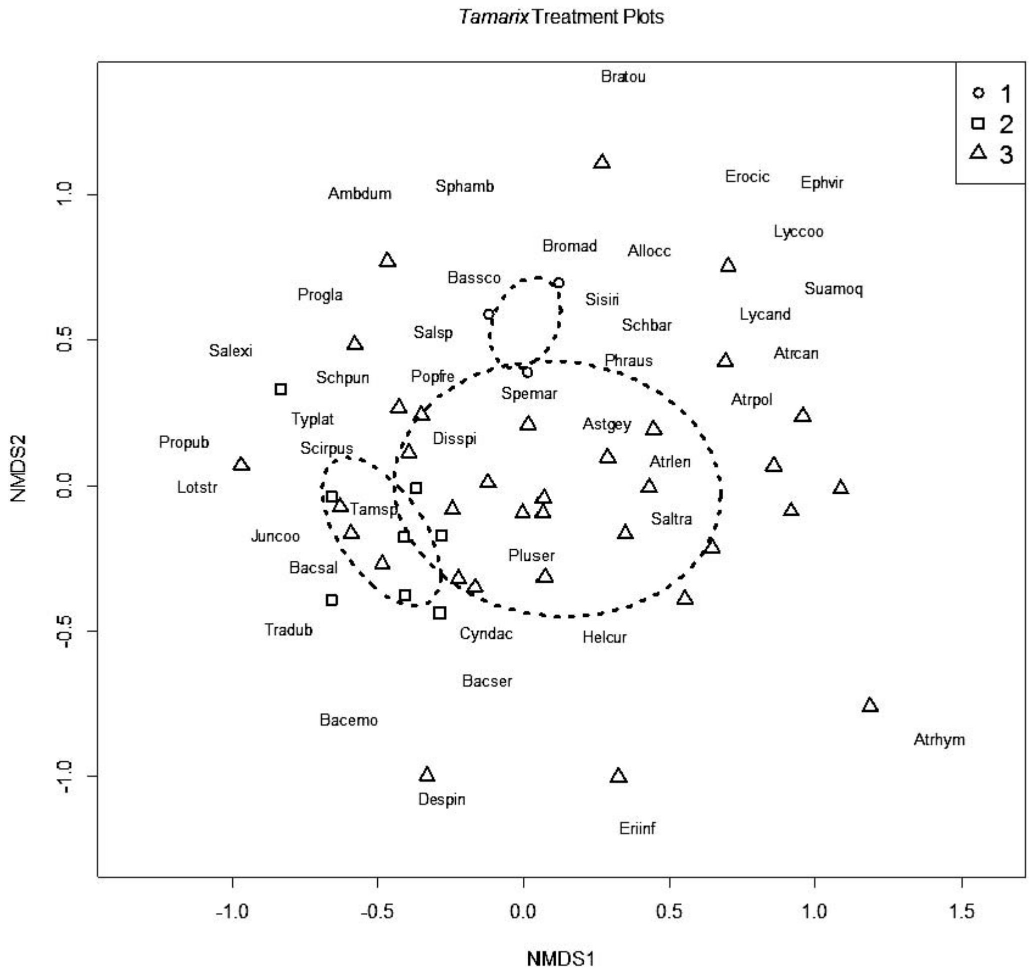

Figure 3. Non metric multidimensional scaling (nMDS) analyses based on a complete linkage cluster analysis using the similarity profile permutation test (SIMPROF) produced three significant groups at $\alpha = 0.05$ (1, 2 and 3) with treatment plots represented as circles (1), squares (2) and triangles (3). The species detected within treatment plots are plotted demonstrating relative dominance with different treatment plots indicating a partitioning of treatment response according to general life history characteristics.

dominant factor, as *Tamarix* spp. is considered a poor competitor in direct interference interactions (Hultine and Dudley 2013). Still, the very low richness values of around two species per 150 m^2 indicates that although reduced *Tamarix* spp. abundance may have driven this diversity response, the combination of soil disturbance and residual 'legacy' effects of *Tamarix* spp. dominance probably inhibited the recovery of associated vegetation to pre-invasion diversity levels. Tamarisk and Russian olive stump cut and whole-plant removal also had little effect on species richness in other riparian systems (Reynolds and Cooper 2011) suggesting that biodiversity recovery from *Tamarix* spp. invasion can be a long and potentially unattainable objective without active intervention in the restoration process.

Patterns of Species Composition within Treated Areas. The nMDS analyses identified three significant vegetation groupings within the treated plots (SIMPROF test $\alpha = 0.05$) (Figure 3). Group one is mainly represented by nonnative herbaceous species including *Bromus* spp. (brome grass species), *Bassia scoparia* (L.) A.J. Scott (kochia), *Sisymbrium irio* L. (London rocket), and *Schismus barbatus* (Loefl. ex. L.) Thell. (common mediterranean-grass). Plots grouped into the second cluster contained higher amounts of wet soil tolerant species such as *Juncus* spp. (rushes), *Typha latifolia* L. (common cattail) and *Cynodon dactylon* (L.) Pers. (bermudagrass). Group three is represented by species more characteristic of drier and more alkaline conditions than might be found in the other two groups including prickly *Salsola tragus* L. (Russian-thistle), *Atriplex lentiformis* (Torr.) S. Watson, (big saltbush), *Pluchea sericea,* and *Distichlis spicata* (L.) Greene (saltgrass).

The presence of different species assemblages within treated areas is an indication of the range of biophysical conditions characterizing areas that *Tamarix* spp. currently occupies in the Virgin River region. Such variability may be dependent on local biotic and abiotic conditions as well as the nature or degree of their disturbance history. It is possible that the efficacy of *Tamarix* spp. removal treatments, the potential for secondary plant invasions, and the recovery potential of native species and biodiversity in general, may all vary among areas differing in biophysical characteristics. However, the current study was not designed to evaluate these factors and there was not enough replication among the three species assemblages to conduct a posthost analysis.

Management Context of the Treatment Effects. Most of the Tamarix spp. control treatments implemented on the Virgin River have been supported by funding for hazardous fuels reduction (BLM, Las Vegas Field Office records), and it appears that fuels reduction was accomplished, at least in the context of the 2- to 5-yr post-treatment context of this study. Reductions in the highly flammable *Tamarix* spp. (Drus 2012) and nonnative annual grasses (D'Antonio and Vitousek 1992), plus overall reductions in woody plant cover, were specific indicators of reduced landscape flammability in this study.

Many *Tamarix* spp. control treatments have also been implemented to promote the recovery of native vegetation, wildlife habitat, and biodiversity in general. In the current study, native woody cover increased (Figure 1B), as did native annual forb density (Figure 2) and most measures of species diversity (Table 1). Density of nonnative annual grasses decreased, but density of nonnative annual forbs increased (Figure 2), indicating both positive and negative responses by nonnatives. We interpret the net effect of all these responses to be generally positive in terms of natural resources management, although there are some very important caveats which are discussed below.

Although there were significant increases in native species density, cover, and species diversity found at treated sites in this study, it remains unknown how close these values are to pre-invasion conditions which might serve as appropriate recovery targets. Some of the response values seem exceedingly small and remain much lower than reasonable target conditions; even though treated and untreated areas were statistically different, small differences may not be ecologically important. An extreme example is exhibited by species richness of woody plants, which was statistically higher in untreated (2.24 species) than treated (1.93 species) areas (Table 1), but these values were based on a relatively large 150 m^2 sampling area and thus the density of woody species per unit area remains very low. Plant community descriptions for pre-invasion riparian woodlands and shrublands are needed to definitively address the ability of any vegetation management treatment to attain the objective of restoring historical or otherwise site-appropriate native vegetation conditions.

Our results do not shed any specific light on the potential vegetation response to the northern tamarisk leaf beetle, *Diorhabda carinulata*, a biological control agent for *Tamarix* spp. which has resulted in mortality rates of $> 70\%$ following multiple years of defoliation in some locations (Bean et al. 2012). Biocontrol can also spread on its own across the landscape, making the effective treatment area much larger than the initial release sites, and potentially interacting with other control methods to enhance target weed mortality (Drus 2012). The leaf beetle established in the Virgin River watershed during the course of this study (see Bateman et al. 2010), although it had not yet dispersed to our study plots by the time that vegetation data were collected. We also cannot compare our results with those that may have been achieved using different control strategies, because this study only reported the results of one mechanical treatment approach. For example, herbicides or the tamarisk leaf beetle can kill *Tamarix* plants, but they typically result in increased dead standing fuels which may increase fire hazard in the short term before dead foliage drops. In addition, the results of this study only reflect results during the first 5 post-treatment yr. Longer-term responses are unknown, but are important in determining how long treatment effects persist and how frequently treatments may need to be reapplied. Different treatments also incur different costs, and cost benefit analyses are often useful in choosing the appropriate *Tamarix* spp. management approach. Finally, the introduction of biological methods of *Tamarix* spp. control may fundamentally change the way managers plan and implement weed suppression, incorporating herbivory as a tactical element in anticipating ecosystem responses (Dudley and DeLoach 2004).

Conclusions. Based on these short-term retrospective results, traditional mechanical methods for reducing abundances of *Tamarix* spp. can provide some significant, but modest, improvement in Mojave Desert riparian vegetation characteristics (species diversity, presence, and relative abundance of native plants). Ecosystem objectives, particularly wildland fuels reduction, may be more successfully met in the short term by such approaches. Wildlife habitat enhancement, although not a direct consideration in this study, may be less successful over the short-term post-treatment periods reflected in this study (2 to 5 yr) if the objectives are to increase architectural structure of vegetation.

Goals of both biodiversity conservation and ecosystem function, including wildfire risk reduction, may be most effectively achieved by targeting aggressive control measures where the need to reduce negative socioeconomic impacts of invasive species outweighs biodiversity goals, while applying more sophisticated approaches where conservation goals are of primary importance. A successful strategy can combine location-specific objectives and ecosystem conditions to plan weed suppression and riparian restoration with a clear understanding of what is achievable in a given timeframe (Shafroth et al. 2008; Taylor and McDaniel 2004).

Acknowledgments

Clark County, NV and the U.S. Geological Survey, Invasives Species Program, provided the funding for this research. Susan Jones and Rebecca Sherry provided helpful comments on earlier versions of this paper. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the US Government.

Literature Cited

Allen EB (1982) Water and nutrient competition between *Salsola kali* and two native grass species (*Agropyron smithii* and *Bouteloua gracilis*). Ecology 63:732–741

Bateman HL, Dudley TL, Bean DW, Ostoja SM, Hultine KR, Kuehn MJ (2010) A river system to watch: documenting the effects of saltcedar (*Tamarix* spp.) biocontrol in the Virgin River Valley. Ecol Restor 28:405–410

Bateman HL, Ostoja SM (2012) Invasive woody plants affect the composition of native lizard and small mammal communities in riparian woodlands. Anim Conserv 15:294–304

Bay RF, Sher AA (2008) Success of active revegetation after *Tamarix* removal in riparian ecosystems of the southwestern United States: A quantitative assessment of past restoration projects. Restor Ecol 16:113–128

Bean DW, Dudley TL, Hultine KR (2012) Bring on the beetles: The history and impact of tamarisk biological control. Pages 377–403 *in* Sher A, Quigley M, eds. *Tamarix*: A case study of ecological change in the American West. New York: Oxford University Press

Beckie HJ, Francis A (2009) The biology of Canadian weeds. 65. *Salsola tragus* L. Can J Plant Sci 89:775–789

Billington DP, Jackson DC, Melosi MV (2005) The history of large federal dams: Planning, design and construction in the era of big dams. USDI-Bureau of Reclamation File Report. 623 p

D'Antonio C, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: A synthesis. Restor Ecol 10: 703–713

D'Antonio CM, Vitousek PM (1992) Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. Annu Rev Ecol Syst 23:63–87

Douglass CH, Nissen SJ, Hart CR (2013) Tamarisk management: lessons and techniques. Pages 333–353 *in* Sher AA, Quigley M, eds. *Tamarix*: A case study of ecological change in the American West. New York: Oxford University Press

Drus GM (2012) Fire ecology of *Tamarix*. Pages 240–255 *in* Sher AA, Quigley M, eds. Tamarix: A case study of ecological change in the American West. New York: Oxford University Press

Drus GM, Dudley TL, Brooks ML, Matchett JR (2012) Influence of biological control on fire intensity profiles of tamarisk (*Tamarix ramosissima* Lebed.). Int J Wildland Fire 22:446–458

Dudley T, Bean D (2012) Tamarisk biocontrol, endangered species risk and resolution of conflict through riparian restoration. BioControl 57:331–347

Dudley TL, Brooks ML (2011) Effectiveness monitoring of springfed wetlands and riparian restoration treatments: progressive management of invasive tamarisk in the southern Nevada Region. Final report, Clark County Desert Conservation Program, Project 2005-UCSB-552-P. 158 p

Dudley TL, DeLoach CJ (2004) Saltcedar (*Tamarix* spp.), endangered species and biological weed control - can they mix? Weed Technol 18: 1542–1551

Fleishman E, McDonal N, MacNally R, Murphy DD, Walters J, Floyd T (2003) Effects of floristics, physiognomy and non-native vegetation on riparian bird communities in a Mojave Desert watershed. J Anim Ecol 72:484–490

Friedman JM, Auble GT, Shafroth PB, Scott ML, Merigliano MF, Freehling MD, Griffin ER (2005) Dominance of non-native riparian trees in western USA. Biol Invasions 7:747–751

Friedman JM, Osterkamp WR, Lewis WM (1996) The role of vegetation and bed-level fluctuations in the process of channel narrowing. Geomorphology 14:341–351

Harms RS, Hiebert RD (2006) Vegetation response following invasive tamarisk (*Tamarix* spp.) removal and implications for riparian restoration. Restor Ecol 14:461–472

Hart CR, White LD, McDonald A, Sheng ZP (2005) Saltcedar control and water salvage on the Pecos river, Texas, 1999–2003. J Environ Manage 75:399–409

Hilmes MM, Vaill JE (1997) Estimates of bridge scour at two sites on the Virgin River, southeastern Nevada using a sediment-transport model and historical geomorphic data. U.S. Geological Survey Water-Resources Investigations Report 97-4073. http://pubs.usgs.gov/wri/wri974073/report.html#HDR28. Accessed January 7, 2013

Horton JS, Campbell CJ (1974) Management of phreatophyte and riparian vegetation for maximum multiple use values. Res. Pap. RM-117. Fort Collins, CO: U.S.D.A. Forest Service, Rocky Mountain Forest and Range Experiment Station. 23 p

Hultine K, Dudley TL (2013) *Tamarix* from organism to landscape. Pages 149–167 *in* Sher A, Quigley M, eds. Tamarix: A case study of ecological change in the American West. New York: Oxford University Press

Jaeger KL, Wohl E (2011) Channel response in a semiarid stream to removal of tamarisk and Russian olive. Water Resour Res 47

Jones ZF, Bock CE (2005) The Botteri's sparrow and exotic Arizona grasslands: an ecological trap or habitat regained? Condor 107:731–741

Mack RN, Foster SK (2009) Eradicating plant invaders: combining ecologically-based tactics and broad-sense strategy. Pages 35–60 *in* Inderjit, ed. Management of Non-native Invasive Plant Species. Heidelberg, Germany: Springer

Marshall RM, Stoleson SH (2000) Chapter 3: Threats. Pages 13–24 *in* Status, ecology, and conservation of the southwestern willow flycatcher. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-60

McDaniel KC, Taylor JP (1999) Steps for restoring bosque vegetation along the middle Rio Grande of New Mexico. Pages 713–714 *in* Eldridge D, Freudenberger D, eds. People and rangelands: building the future: Proceedings, 6th International Rangeland Congress. Aitkenvale, Queensland, Australia: Townsville, Queensland

Meinhardt KA, Gehring CA (2013) *Tamarix* and soil ecology. Pages 225–239 *in* Sher A, Quigley M, eds. *Tamarix*: A case study of ecological change in the American West. New York: Oxford University Press

Mortenson SJ, Weisberg PJ (2010) Does river regulation increase dominance of woody species in riparian landscapes? Global Ecol Biogeogr 19:562–574

Naiman RJ, Decamps H, Pollock M (1993) The role of riparian corridors in maintaining regional biodiversity. Ecol Appl 3:209–212

Oksanen J, Blanchet FG, Kindt R, Legendre P, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Wagner H (2012) vegan: Community Ecology Package. R package version 2.0-5. http://CRAN.R-project.org/package=vegan. Accessed March 12, 2013

O' Meara S, Larsen D, Owens C (2010) Methods to control saltcedar and Russian olive. Pages 65–102 *in* Shafroth PB, Brown CA, Merritt DM, eds. Saltcedar and Russian olive control demonstration act science assessment: U.S. Geological Survey Scientific Investigations Report 2009–5247

Parsons AJ, Abrahams AD, Wainwright J (1996) Responses of interrill runoff and erosion rates to vegetation change in southern Arizona. Geomorphology 14:311–317

Patten DT (1998) Riparian ecosystems of semi-arid North America: Diversity and human impacts. Wetlands 18:498–512

R Core Team (2012) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0. http://www.R-project.org/. Accessed March 12, 2013

Reynolds LV, Cooper DJ (2011) Ecosystem response to removal of exotic riparian shrubs and a transition to upland vegetation. Plant Ecol 212:1243–1261

Richardson DM, Holmes PM, Esler KJ, Galatowitsch SM, Stromberg JC, Kirkman SP, Pysek P, Hobbs RJ (2007) Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Divers Distrib 13:126–139

Robinson TW (1965) Introduction, spread and areal extent of saltcedar (*Tamarix*) in the western states. USGS Prof. Paper 491-A

Sabo JL, Sponseller R, Dixon M, Gade K, Harms T, Heffernan J, Jani A, Katz G, Soykan C, Watts J, Welter J (2005) Riparian zones increased regional species richness by harboring different, not more species. Ecology 86:56–62

SAS Institute, Inc (2009) JMP 8.0 Statistical Discovery Software. SAS Institute, Inc., Cary, NC, USA

Seaby RM, Henderson PA (2006) Species Diversity and Richness Version 4. Lymington, England: Pisces Conservation

Shafroth PB, Beauchamp VB, Briggs MK, Lair K, Scott ML, Sher AA (2008) Planning riparian restoration in the context of *Tamarix* control in western North America. Restor Ecol 16:97–112

Shafroth PB, Brown CA, Merritt DM, eds. (2010) Saltcedar and Russian olive control demonstration act science assessment: U.S. Geological Survey Scientific Investigations Report 2009–5247. 143 p

Shafroth PB, Cleverly JR, Dudley TL, Taylor JP, van Riper C, Weeks EP, Stuart JN (2005) Control of *Tamarix* in the western United States — implications for water salvage, wildlife use and riparian restoration. Environ Manage 35:1–16

Sogge MK, Felley DL, Wotawa M (2005) A quantitative model of avian community and habitat relationships along the Colorado River in the Grand Canyon. Pages 161–192 *in* van Riper C, Mattson D, eds. The Colorado Plateau II: biophysical, socioeconomic and cultural research. Tucson, AZ: University of Arizona Press

Solow AR (1993) A simple test for change in community structure. J Anim Ecol 62:191–193

Stromberg JC (2001) Restoration of riparian vegetation in the south-western United States: importance of flow regimes and fluvial dynamism. J Arid Environ 49:17–34

Taylor JP, McDaniel KC (1998) Restoration of saltcedar (*Tamarix* sp.) -infested floodplains on the Bosque del Apache National Wildlife Refuge. Weed Technol 12:345–352

Taylor JP, McDaniel KC (2004) Revegetation strategies after saltcedar (*Tamarix* spp.) control in headwater, transitional, and depositional watershed areas. Weed Technol 18:1278–1282

van Riper C, Paxton KL, O'Brien C, Shafroth PB, McGrath LJ (2008) Rethinking avian response to *Tamarix* on the lower Colorado River: a threshold hypothesis. Restor Ecol 16:155–167

Vincent KR, Friedman JM, Griffin ER (2009) Erosional consequence of saltcedar control. Environ Manage 44:218–227

Whitaker D, Christman M (2010) Clustsig: Significant Cluster Analysis. R package version 1.0. http://CRAN.R-project.org/package=clustsig. Accessed March 12, 2013

Young JA (1991) Tumbleweed. Sci Am 264:82–87

Received September 5, 2013 and accepted February 12, 2014.